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Effects of maize straw and its biochar application on organic and humic carbon in water-stable aggregates of a Mollisol in Northeast China: A five-year field experiment

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ABSTRACT

In agricultural ecosystems, one optimized crop residue management practice, i.e., incorporating biochar produced from crop residues via pyrolysis to the soil, is considered a promising technology for mitigating climate change and improving soil quality. However, the potential mechanisms are still not well known. In this study, a five-year (2011–2015) field experiment in a Mollisol of Northeast China that included four treatments, i.e., CK (no fertilizer), NPK (mineral nitrogen, phosphorus and potassium fertilizers), NPKS (NPK plus maize straw) and NPKB (NPK plus maize straw-derived biochar), was conducted to evaluate the effects of maize straw and its derived biochar application on the following: 1) the size distribution and water stability of soil aggregates; 2) the concentrations of organic and humic carbon (C) in bulk soil and water-stable aggregate fractions; and 3) the chemical compositions of organic and humic C in water-stable aggregate fractions. Compared with the CK and NPK treatments, the proportion of macroaggregates (> 0.25 mm) [mainly large macroaggregates (> 2 mm)] increased whereas the proportions of microaggregates (0.25–0.053 mm) and silt + clay fractions (< 0.053 mm) decreased in NPKS and NPKB treatments, resulting in the mean weight diameter (MWD) of soil aggregates being higher under NPKS and NPKB treatments than under CK and NPK treatments. The concentrations of organic and humic C in bulk soil and aggregate fractions were also generally higher under NPKS and NPKB treatments than under CK and NPK treatments. The treatment effects were more apparent for NPKB than for NPKS. With respect to other treatments, the NPKB treatment significantly increased the ratios of macroaggregate- to microaggregate-associated organic C as well as humin C. The solid-state ^{13}C cross-polarization magic-angle-spinning (CPMAS) NMR spectra of small macroaggregates (2–0.25 mm) and its humic acid fraction showed that the NPKS treatment did not alter the proportions of the C functional groups. In contrast, the NPKB treatment clearly decreased the proportions of alkyl C and O-alkyl C but increased the proportion of aromatic C, resulting in the alkyl C/O-alkyl C and hydrophobic C/hydrophilic C ratios being higher whereas the aliphatic C/aromatic C ratio was lower under NPKB than under other treatments. These findings indicate that the application of maize straw biochar enhanced soil aggregate stability and aggregate-associated organic and humic C concentrations. Biochar C was preferentially incorporated into macroaggregates and its humin fraction with respect to microaggregates. Moreover, the molecular structure of small macroaggregates and its HA fraction became more stable, hydrophobic and aromatic. Therefore, physical protection through organic C in aggregate fractions, physical-chemical protection through the encapsulation of humic C within aggregate fractions, and physical-biochemical protection through recalcitrant alkyl C and aromatic C within aggregate fractions may be the primary mechanisms for C preservation in biochar-amended soil. Our results suggested that maize straw biochar application was an effective strategy based on the comprehensive goal of mitigating climate change and improving soil quality in Mollisols of Northeast China.

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1. Introduction

Due to the increasing concentrations of atmospheric greenhouse gases, especially carbon dioxide (CO_2), the global mean surface temperature has risen by 0.72 °C over the past 60 years (1951–2012) (IPCC, 2014). Soil is the largest carbon (C) reservoir in terrestrial ecosystems, storing approximately 1500 Pg of organic C in the top 1 m (Batjes, 1996). Even small changes in soil organic carbon (SOC) stocks can have a substantial influence on atmospheric CO_2 concentration (Han et al., 2016). Moreover, SOC plays a crucial role in the maintenance of soil productivity due to its significant contribution to physical, chemical and biological properties of soil (Lal et al., 2013). Therefore, it is important to optimize strategies to enhance SOC sequestration due to its relevance to climate change mitigation and soil sustainability (Stockmann et al., 2013).

In agroecosystems, returning crop residue to soil is a common practice to improve SOC sequestration (Chen et al., 2016; Ghimire et al., 2017; Zhao et al., 2018a). In recent years, one optimized crop residue management measure, i.e., incorporating biochar produced from residues via pyrolysis to the soil, has already received extensive attention (Ding et al., 2017). Owing to the particular properties of biochar, such as its richness in C, high stability, and large surface area (Tan et al., 2017), it is considered a promising material for both climate change mitigation (Mohammadi et al., 2016) and soil quality improvement (Agegnehu et al., 2017). However, most prior results were obtained through laboratory incubation or pot experiments. Studies regarding biochar's impact on soil processes through field experiments, especially long-term ones, are still lacking (Ding et al., 2017). This limits our understanding of the potential mechanisms at work in soil processes after biochar application.

The formation and stability of soil aggregates is a critical process for the sequestration of C in soil (Six et al., 2002). SOC encapsulated within soil aggregates can be physically protected against microbial attacks. As a chemically resistant SOC fraction, humic substances are considered to be a persistent cementing agent involved in the formation and stabilization of soil aggregates (Bronick and Lal, 2005; Spaccini et al., 2006). On the other hand, the persistence of humic substances can be enhanced by their occlusion in soil aggregates (Bongiovanni and Lobartini, 2006). Crop residues are one of the major sources of humic substances in farmland soil. It is estimated that if 15% C assimilated in crop residues were converted to humic substances, global C sequestration would be approximately 5.0 Pg by 2020 (Lal, 1997). Therefore, examining the responses of organic and humic C associated with soil aggregates to crop residues-derived biochar can help us gain a deeper understanding of the mechanisms of C sequestration in biochar-amended soil.

Northeast China is the most productive maize-growing area in China (Wang et al., 2014). It is estimated that the total amount of maize straw in Northeast China was approximately 1.19×10^{11} kg in 2015 (Shi et al., 2017). Therefore, the main purposes of the present study are to examine the effects of a five-year application of maize straw and its biochar on (1) the soil aggregate size distribution and water stability of soil aggregates, (2) the concentrations of organic and humic C in bulk soil and its water-stable aggregate size fractions, and (3) the chemical compositions of organic and humic C in soil water-stable aggregate size fractions. Because the C concentrations and chemical compositions of humic-like substances derived from maize straw biochar were clearly different from those derived from maize straw (Zhang et al., 2015a,b), we hypothesize that maize straw biochar application to soil may have distinct impacts on the quantity and quality of organic and humic C as well as the aggregate stability compared to maize straw application.

2. Materials and methods

2.1. Study site, experimental design and soil sampling

A field experiment with continuous maize (*Zea mays* L.) cropping was initiated in April 2011 at the Jilin Academy of Agricultural Sciences (42°40′N, 124°88′E), Gongzhuling City, Jilin Province, Northeast China. The climate is temperate continental monsoon, with a mean annual temperature of 5.6 °C and a mean annual precipitation of 600 mm. The soil is classified as Mollisol (USDA soil Taxonomy) with clay loam texture. The initial soil properties in the 0–20 cm layer were as follows: pH, 5.7; organic carbon (C), 13.3 g kg^{-1}; total nitrogen (N), 1.39 g kg^{-1}; alkali-hydrolysable N, 130.4 mg kg^{-1}; Olsen phosphorus (P), 176.1 mg kg^{-1}; and ammonium acetate-extractable potassium (K), 7.50 mg kg^{-1}. The maize straw biochar used in the experiment was produced via slow pyrolysis of maize straw at Beijing United Pioneer Environmental Engineering Co., Ltd., China (Zhang et al., 2015a,b).

The experiment was arranged in a completely randomized block design with three replicates. Each experimental plot was 104 m^2 (10.4 m × 10 m). Four treatments including unfertilized control (CK), mineral nitrogen, phosphorus and potassium fertilizers (NPK), NPK plus maize straw (NPKS), and NPK plus maize straw biochar (NPKB) were included in the present study. The mineral fertilizers were applied at 225.0 kg N ha^{-1}, 82.5 kg P_2O_5 ha^{-1}, and 82.5 kg K_2O ha^{-1} per year in the form of urea, multiple superphosphate, and potassium sulfate, respectively. The maize straw (containing 432.9 g C kg^{-1}) and maize straw biochar (containing 597.2 g C kg^{-1}) were applied based on equal total C (i.e., 3200 kg C ha^{-1} per year). All multiple superphosphate and potassium sulfate and 40% of urea fertilizers were applied at seeding time, and the remaining urea was applied in early- to mid-June. Maize straw and its biochar were applied 10 cm below the soil surface after maize harvesting each year. Maize was sown in late April and harvested in early October. After harvesting the grain, all aboveground crop residues were removed from the field except for the NPKS plot.

Surface soil samples (0–10 cm depth) were collected after the maize harvest in October 2015. In each of the three replicate plots, five randomly selected soil cores were taken and mixed as a composite sample. The fresh soil samples were sieved to pass through a 10 mm sieve and then air dried. One portion of each sample was used for soil aggregates separation, and the other was sieved to 2 mm for soil properties analyses.

2.2. Soil analysis

2.2.1. Soil aggregates isolation

Soil water-stable aggregates were isolated via the wet sieving method (Cambardella and Elliott, 1993) into four size classes, namely, large macroaggregates (> 2 mm), small macroaggregates (2–0.25 mm), microaggregates (0.25–0.053 mm) and silt + clay fractions (< 0.053 mm).

2.2.2. Extraction, fractionation and purification of humic substance fractions

Humic substance fractions were extracted, fractionated and purified according to the procedure described previously (Zhang et al., 2010; Wang et al., 2016). Briefly, bulk soil and soil aggregate samples were extracted using a 0.1 M NaOH and 0.1 M $Na_4P_2O_7$ solution. The alkaline supernatants were acidified with 6 M HCl to precipitate the acid-insoluble humic acid (HA) from acid-soluble fulvic acid (FA). The alkali-insoluble solid residue was humin (HU).

For ^{13}C NMR analysis, HA was further purified by deashing with 0.5% HCl-HF solution after three cycles of dissolution in 0.1 M NaOH and flocculation in 6 M HCl HU was purified by deashing with 10% HCl-HF solution. Both HA and HU were dialyzed against distilled water, freeze-dried, and ground to < 0.1 mm. In this study, only HA and HU from small macroaggregates were prepared for the ^{13}C NMR analysis, as

this aggregate size class was the most abundant (40.8–46.5% of the total soil mass).

2.2.3. Organic and humic C measurements

Organic C concentrations in bulk soil and its aggregate fractions were measured using $K_2Cr_2O_7$-H_2SO_4 oxidation at 170–180 °C followed by titration with 0.1 M $FeSO_4$ (Lu, 2000). Humic acid carbon (HAC) and fulvic acid carbon (FAC) were measured using a Shimadzu TOC-VCPH analyzer, and humin carbon (HUC) was calculated by subtracting the sum of HAC and FAC from SOC (Li et al., 2017a,b,c).

2.2.4. Solid-state ^{13}C NMR analysis

Solid-state ^{13}C cross-polarization magic-angle-spinning (CPMAS) NMR spectra of small macroaggregates and its HA and HU samples were recorded on a Bruker AVANCE III 400 WB spectrometer at 100.6 MHz, according to the method reported previously (Wang et al., 2016). The spectra were divided into four common chemical shift regions representing alkyl C (0–50 ppm), O-alkyl C (50–110 ppm), aromatic C (110–160 ppm), and carbonyl C (160–210 ppm). Prior to solid-state ^{13}C NMR analysis, small macroaggregate samples were pretreated with 10% HCl-HF solution, as recommended by Schmidt et al. (1997), to improve the signal-to-noise ratio of the resultant NMR.

2.3. Calculation and statistical analysis

The mean weight diameter (MWD), which was used as indices to evaluate soil aggregate stability, was calculated as follows (Gupta Choudhury et al., 2014:

$$\mathrm{MWD(mm)} = \frac{\sum_{i=1}^{n} X_i W_i}{\sum_{i=1}^{n} W_i}$$

where n is the number of aggregate fractions, X_i is the mean diameter of each fraction class, and W_i is the weight percentage of each fraction class.

The ratios of alkyl C to O-alkyl C (alkyl C/O-alkyl C, A/O-A), aliphatic C to aromatic C (aliphatic C/aromatic C, Alip/Arom) and hydrophobic C to hydrophilic C (hydrophobic C/hydrophilic C, HB/HI), which were used as indices to assess the degrees of stability, aliphaticity and hydrophobicity of soil organic matter, respectively, were calculated as follows (Zhang et al., 2011; Li et al., 2014)):

$$\mathrm{A/O-A} = \frac{\mathrm{AlkylC}}{\mathrm{O\text{-}alkylC}}$$

$$\mathrm{Alip/Arom} = \frac{\mathrm{AlkylC+O-alkylC}}{\mathrm{AromaticC}}$$

$$\mathrm{HB/HI} = \frac{\mathrm{Alkyl\ C+\ Aromatic\ C}}{\mathrm{O-alkylC+CarbonylC}}$$

Statistical analysis was conducted using SPSS 16.0 software. The differences among treatment means were detected using one-way analysis of variance (ANOVA) followed by the least significant difference (LSD) test at $P < 0.05$.

3. Results

The maize grain yields during 2011–2015 are presented in Fig. 1. In general, maize yields were higher under NPKS and NPKB than under CK and NPK treatments, with significant differences between CK and NPK treatments with NPKS treatment in 2012 and between CK treatment with other treatments in 2013 and 2014. In 2015, maize yield significantly increased in the order CK < NPK < NPKB < NPKS. The lower maize yield in 2015 compared to other years could be due to the extreme weather (heat and drought) occurring then.

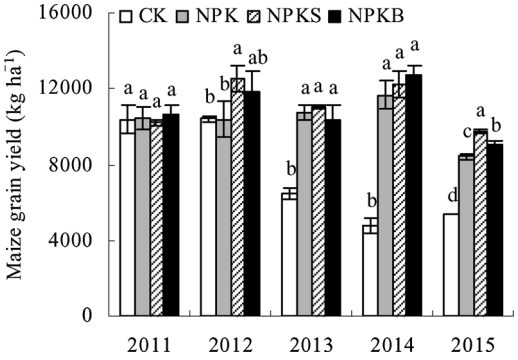

Fig. 1. Maize grain yield during the period 2011–2015 under treatments receiving mineral fertilizers (NPK), mineral fertilizers plus maize straw (NPKS), mineral fertilizers plus maize straw-derived biochar (NPKB), and no fertilizers (CK). Error bars represent the standard deviations of the mean (n = 3). Different lowercase letters indicate significant differences among treatments in each year ($P < 0.05$).

3.1. Concentrations of organic and humic carbon in bulk soil

After five years of the field experiment, the SOC concentration increased in the order CK < NPK < NPKS < NPKB (Fig. 2a). No significant difference was observed between CK and NPK treatments.

Similar to the SOC concentration, the HAC and HUC concentrations also increased in the order CK < NPK < NPKS < NPKB (Fig. 2b). There were neither significant differences for HAC between NPK and NPKS treatments nor for HUC between CK and NPK and between NPKS and NPKB treatments. In contrast, the FAC concentration increased in the order NPK < NPKS < CK < NPKB, with a significant difference between NPK and NPKB treatments.

The HAC/FAC ratio was significantly lower under CK treatment than under other treatments (Fig. 2c). However, no significant difference was found among the NPK, NPKS and NPKB treatments. Moreover, the (HAC + FAC)/HUC ratio was not significantly different among all of the treatments (Fig. 2d).

3.2. Soil aggregate size distribution and aggregate stability

Across all treatments, the proportion of small macroaggregates was the highest, followed by that of microaggregates (Fig. 3a). Compared with CK and NPK treatments, the NPKS and NPKB treatments increased the proportion of macroaggregates whereas decreased the proportions of microaggregates and silt + clay fractions. There were significant differences between CK and NPK treatments with NPKB treatment for large macroaggregates and microaggregates, between CK treatment with NPK, NPKS and NPKB treatments for small macroaggregates, and between CK and NPKB treatment for silt + clay fractions.

The ratios of macroaggregates to microaggregates and large macroaggregates to small macroaggregates were higher under the NPKS and NPKB treatments than under the CK and NPK treatments (Fig. 3b and c). The differences between the NPKB treatment with CK and NPK treatments were significant.

The MWD were higher under the NPKS and NPKB treatments than under the CK and NPK treatments (Fig. 3d). Significant differences were observed between the NPKB treatment and other treatments.

3.3. Concentrations of organic and humic carbon in soil water-stable aggregates

3.3.1. Organic carbon concentration

Within each aggregate size fraction, the organic C concentrations increased in the order CK < NPK < NPKS < NPKB (Fig. 4a). Significant differences were observed between the NPKB treatment and other treatments. Moreover, the small macroaggregate-associated

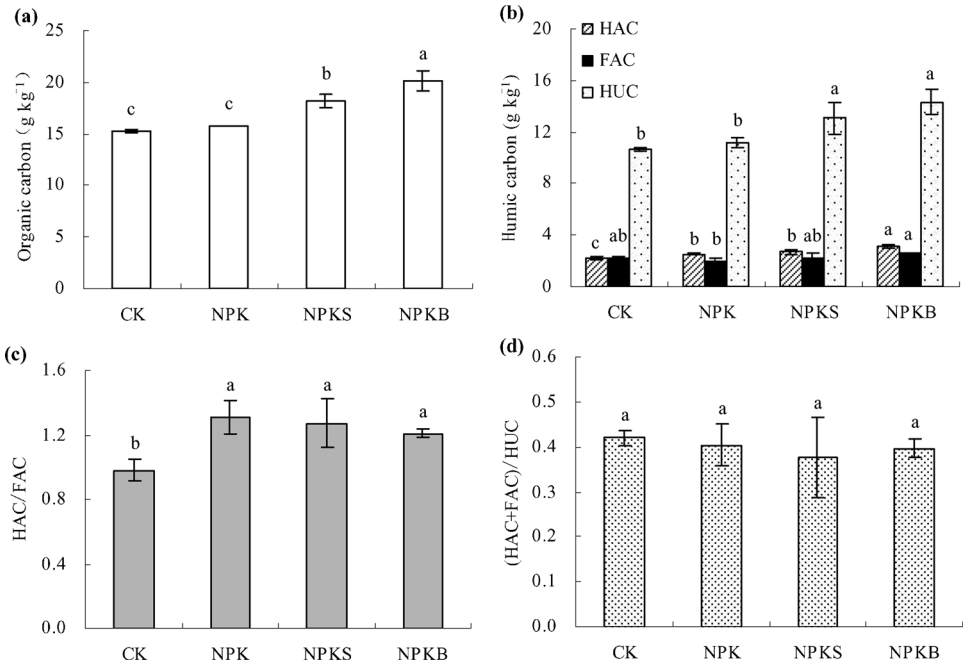

Fig. 2. Concentrations of organic carbon, humic acid carbon (HAC), fulvic acid carbon (FAC) and humin carbon (HUC) and ratios of HAC to FAC (HAC/FAC) and HAC + FAC to HUC [(HAC + FAC)/HUC] in bulk soil under treatments receiving mineral fertilizers (NPK), mineral fertilizers plus maize straw (NPKS), mineral fertilizers plus maize straw-derived biochar (NPKB), and no fertilizers (CK). Error bars represent the standard deviations of the mean (n = 3). Different lowercase letters indicate significant differences among treatments ($P < 0.05$).

organic C concentration was significantly higher under the NPKS treatment than under the CK and NPK treatments, and the microaggregate-associated organic C concentration was significantly higher under the NPKS than under the CK treatment.

The ratio of macroaggregate- to microaggregate-associated organic C also increased in the order CK < NPK < NPKS < NPKB (Fig. 4b). There was no significant difference between NPK and NPKS.

3.3.2. Humic carbon concentration

In general, the HAC, FAC and HUC concentrations within each aggregate size fraction also increased in the order CK < NPK < NPKS < NPKB (Fig. 5a, b and c). The HAC associated with large macroaggregates, small macroaggregates and silt + clay fractions were significantly lower under CK treatment than under other treatments. The FAC associated with microaggregates was significantly lower under CK than under NPKB treatment. The HUC in all aggregate size fractions was significantly higher under NPKB treatment than under other treatments.

The HAC/FAC ratio was significantly higher under NPKB treatment than under other treatments within large macroaggregates. The ratio was also higher under the NPKB than under the CK treatment in small macroaggregates (Fig. 5d). However, no significant differences were observed for microaggregates and silt + clay fractions among the four treatments. In contrast, the (HAC + FAC)/HUC ratio was lower under the NPKB treatment than under other treatments for all aggregate size fractions, with significant differences observed for large and small macroaggregates (Fig. 5e).

The ratio of macroaggregate- to microaggregate-associated HAC

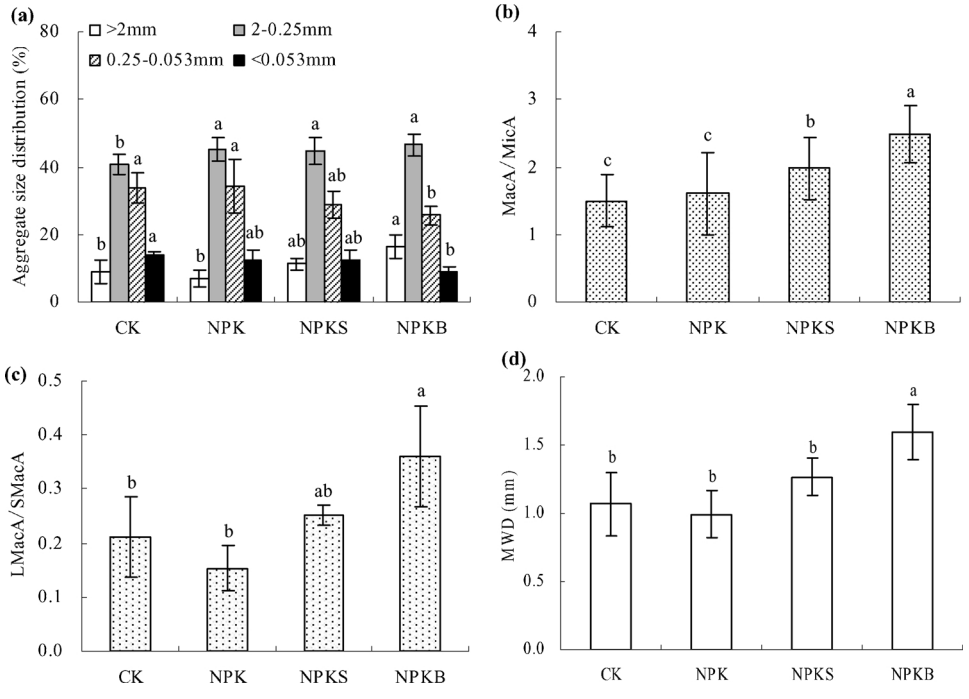

Fig. 3. Size distribution, ratios of macroaggregates to microaggregates (MacA/MicC) and large macroaggregates to small macroaggregates (LMacA/SMacC), and mean weight diameter (MWD) of water-stable aggregates in soil under treatments receiving mineral fertilizers (NPK), mineral fertilizers plus maize straw (NPKS), mineral fertilizers plus maize straw-derived biochar (NPKB), and no fertilizers (CK). Error bars represent the standard deviations of the mean (n = 3). Different lowercase letters indicate significant differences among treatments ($P < 0.05$).

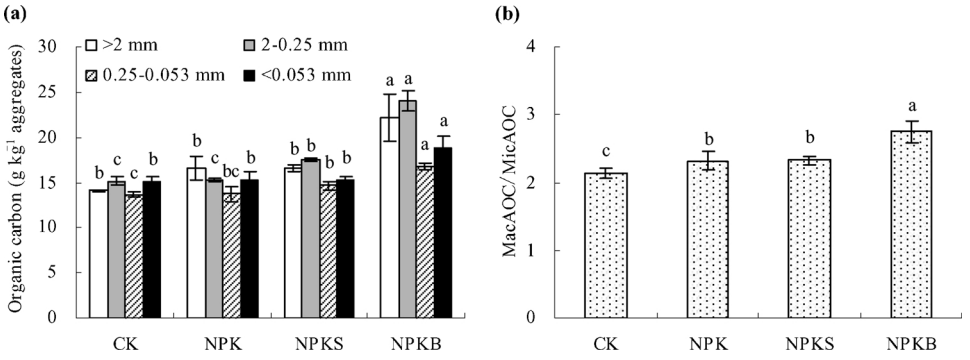

Fig. 4. Organic carbon concentration and ratio of macroaggregate- to microaggregate-associated organic C (MacAOC/MicAOC) in soil water-stable aggregates under treatments receiving mineral fertilizers (NPK), mineral fertilizers plus maize straw (NPKS), mineral fertilizers plus maize straw-derived biochar (NPKB), and no fertilizers (CK). Error bars represent the standard deviations of the mean (n = 3). Different lowercase letters indicate significant differences among treatments ($P < 0.05$).

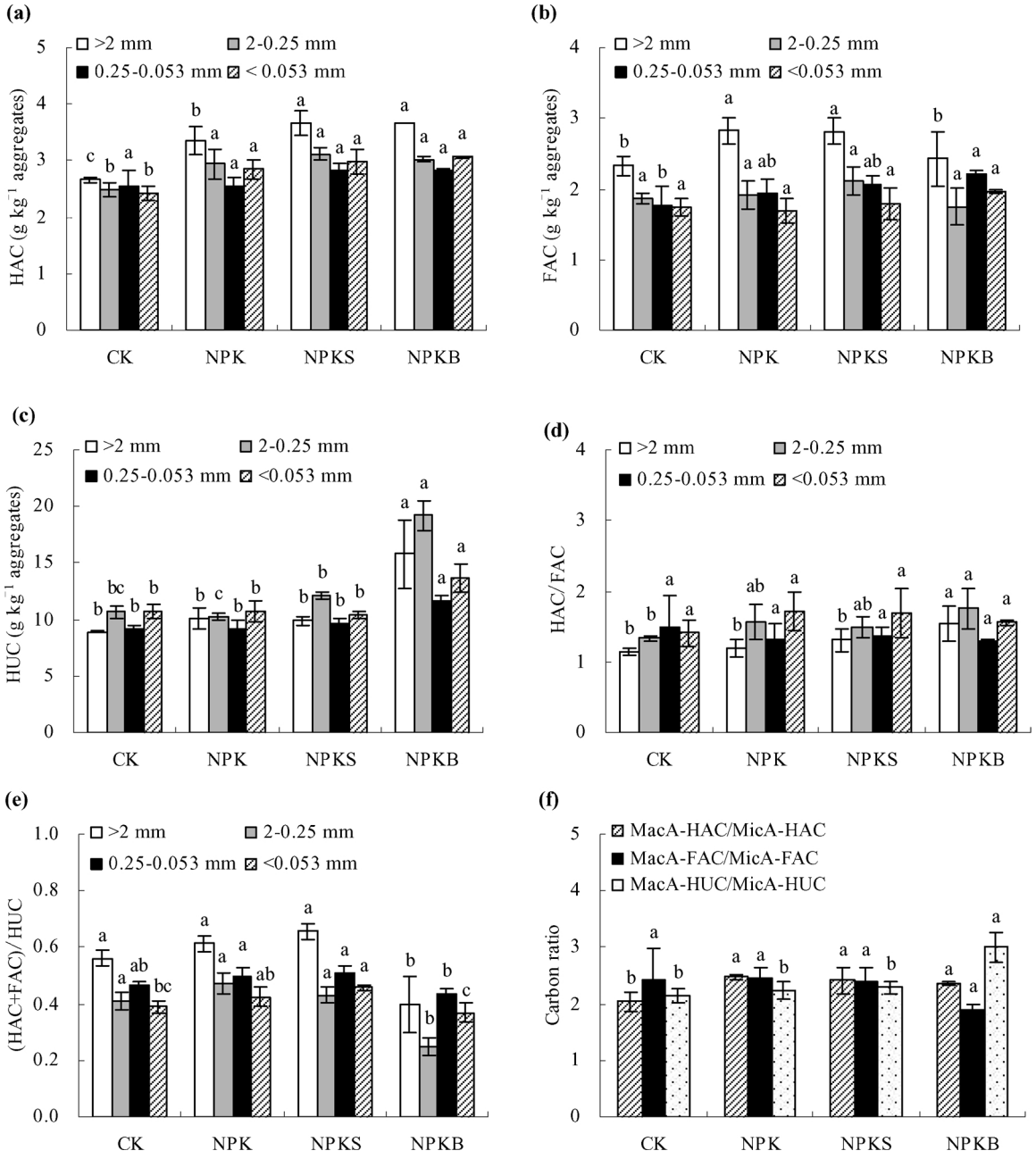

Fig. 5. Concentrations of humic acid carbon (HAC), fulvic acid carbon (FAC) and humin carbon (HUC) and ratios of HAC to FAC (HAC/FAC), HAC + FAC to HUC [(HAC + FAC)/HUC], macroaggregate- to microaggregate-associated HAC (MacA-HAC/MicA-HAC), macroaggregate- to microaggregate-associated FAC (MacA-FAC/MicA-FAC) and macroaggregate- to microaggregate-associated HUC (MacA-HUC/MicA-HUC) in soil water-stable aggregates under treatments receiving mineral fertilizers (NPK), mineral fertilizers plus maize straw (NPKS), mineral fertilizers plus maize straw-derived biochar (NPKB), and no fertilizers (CK). Error bars represent the standard deviations of the mean (n = 3). Different lowercase letters indicate significant differences among treatments ($P < 0.05$).

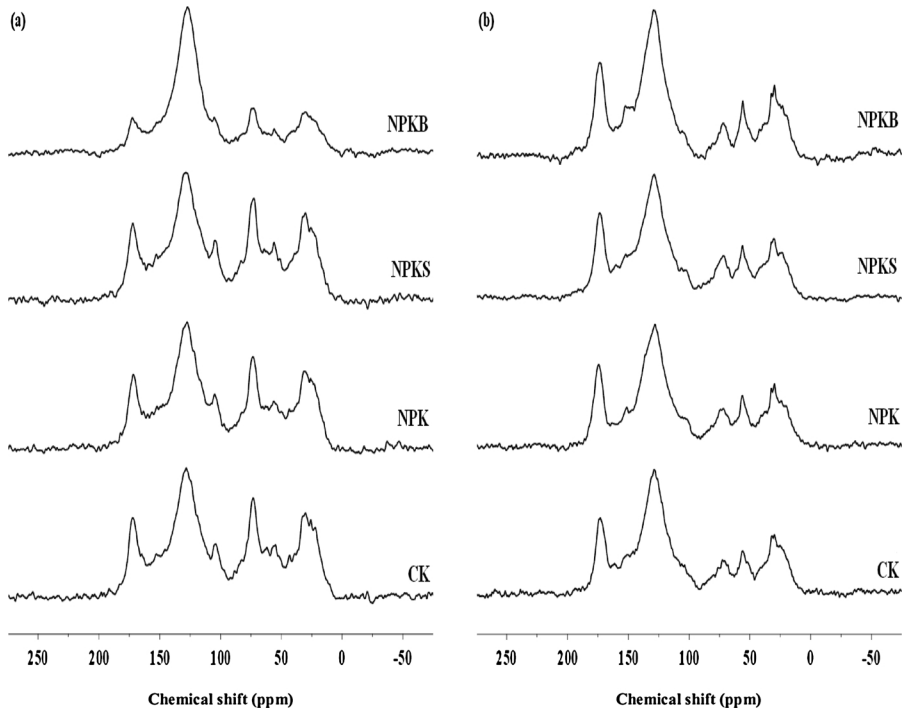

Fig. 6. Solid-state ^{13}C CPMAS NMR spectra of soil small water-stable macroaggregates (a) and its humic acid fraction (b) under treatments receiving mineral fertilizers (NPK), mineral fertilizers plus maize straw (NPKS), mineral fertilizers plus maize straw-derived biochar (NPKB), and no fertilizers (CK).

was significantly lower under CK than under other treatments (Fig. 5f). Furthermore, the ratio of macroaggregate- to microaggregate-associated HUC was significantly higher under NPKB than under other treatments (Fig. 5f).

3.4. Solid-state ^{13}C NMR spectra of soil small water-stable macroaggregates and its humic acid fractions

It was unexpected that the solid-state ^{13}C NMR spectra of HU samples after the application of maize straw biochar were not detectable in the present study. The reason may be due to the presence of static interference, as hypothesized in a previous study (Zhang et al., 2015a,b). Thus, we only reported the solid-state ^{13}C NMR spectra of small water-stable macroaggregates and its HA fractions in this study.

The solid-state ^{13}C CPMAS NMR spectra of all samples exhibited major resonance peaks at approximately 26, 30, 32, 56, 73, 105, 127, 153, and 172 ppm (Fig. 6). The peak assignments were reported in our previous work (Wang et al., 2016; Li et al., 2017a,b,c).

Generally, no evident differences could be observed for the proportions of the C functional groups in both small water-stable macroaggregates and its HA fractions among the CK, NPK and NPKS treatments (Table 1). In contrast, the NPKB treatment clearly changed the proportion of the C functional groups in small water-stable macroaggregates and its HA fractions. Compared with other treatments, the NPKB treatment had lower proportions of alkyl C and O-alkyl C but higher proportion of aromatic C in both small water-stable macroaggregates and its HA fractions. The alkyl C/O-alkyl C and hydrophobic C/hydrophilic C ratios were higher, whereas the aliphatic C/aromatic C ratio was lower under the NPKB treatment than under the other treatments.

4. Discussion

4.1. Effects of mineral fertilizers alone on soil organic and humic carbon concentrations and aggregate stability

As reported in previous studies (Yang et al., 2003; Chen et al., 2010; Lou et al., 2011), the application of mineral fertilizers alone was not adequate to maintain C level under conventional management without aboveground crop residues being returned to soil. Therefore, mineral fertilizers alone generally did not significantly affect organic and humic C concentrations in both bulk soil (Fig. 2a and b) and its water-stable aggregate fractions (Figs. 4a; 5 a–c) compared with the CK treatment. The exception was for HAC, which was significantly higher under mineral fertilizers alone than under CK treatment in bulk soil (Fig. 2b) and macroaggregates and silt + clay fractions (Fig. 5a). This implied that mineral fertilizers alone were beneficial to the formation of HA and that the HA formed had preferentially accumulated in macroaggregates and silt + clay fractions. Because mineral fertilizers alone caused only a slight change in the concentration of SOC, which was a major binding agent involved in the soil aggregation process (Mikha and Rice, 2004; Bi et al., 2015; Li et al., 2017a,b,c), the water stability of soil aggregates was not significantly different from that of the CK treatment (Fig. 3d). The insignificant influence of mineral fertilizers alone on soil aggregate stability has also been reported by other researchers (Wang et al., 2011; Peng et al., 2015; Zhang et al., 2016).

4.2. Effects of maize straw and its biochar application on organic and humic carbon concentrations in bulk soil

The application of maize straw and its biochar in combination with mineral fertilizers significantly increased SOC concentration compared with mineral fertilizers alone (Fig. 2a), which agreed with previous findings from field experiments in Mollisols of Northeast China (Ma et al., 2016; Yang et al., 2017) as well as in other soil (Xie et al., 2013; Zhang et al., 2017; Zhu et al., 2017). The amount of organic C sequestered in soil depended on the balance between C inputs and outputs (Anikwe, 2010). Under equal total C input (i.e., 3200 kg C ha^{-1} per year) in the present study, the significantly higher SOC concentration under maize straw biochar application than under maize straw application (Fig. 2a) could be attributed to the high stability of biochar C with respect to its raw feedstock (Fischer and Glaser, 2012; Tan et al., 2017).

Humic-like substances in organic amendments have been shown to

Table 1
Relative proportions (%) of different carbon functional groups in soil small water-stable macroaggregates and its humic acid fractions obtained from solid-state ^{13}C CPMAS NMR spectra.

Treatment	Alkyl C(0-50 ppm)	O-alkyl C(50-110 ppm)	Aromatic C(110-160 ppm)	Carbonyl C(160-210 ppm)	A/O-A	Alip/Arom	HB/HI
Soil small water-stable macroaggregates							
CK	20.3	29.4	37.5	12.8	0.86	1.17	1.58
NPK	19.8	30.2	37.3	12.7	0.83	1.17	1.56
NPKS	20.4	30.3	36.7	12.5	0.84	1.22	1.55
NPKB	16.2	21.2	52.6	10.1	0.91	0.65	2.47
Humic acid from soil small water-stable macroaggregates							
CK	18.6	20.3	45.3	15.8	0.91	0.70	2.19
NPK	18.5	22.0	44.9	14.6	0.84	0.71	2.26
NPKS	17.4	22.3	45.4	14.9	0.78	0.70	2.15
NPKB	16.9	13.8	53.4	15.9	1.22	0.45	3.06

A/O-A: alkyl C/O-alkyl C.
Alip/Arom: aliphatic C/aromatic C.
HB/HI: hydrophobic C/hydrophilic C.

contribute to the formation of native soil humic substances (Adani and Ricca, 2004; Brunetti et al., 2007; Senesi et al., 2007). Both maize straw and its biochar contained a certain amount of humic-like substance fractions (Adani and Ricca, 2004; Graber et al., 2015; Zhang et al., 2015a,b), the incorporation of which into the soil led to the increase of soil humic C concentrations (Fig. 2b). Hayes (2013) suggested that the HA-like fraction from biochar was produced from the biological oxidation of peripheral components of biochar in soil. Although HA- and FA-like C were higher in maize straw than in its biochar (Zhang et al., 2015a,b), HAC and FAC were lower in the soil under maize straw application than under its biochar application (Fig. 2b). This suggested that humic-like substance fractions in maize straw biochar decomposed slower than those in maize straw. It has been previously reported that the HA-like fraction from maize straw biochar contained larger proportions of alkyl C and aromatic C but smaller proportion of O-alkyl C than that from maize straw (Zhang et al., 2015a,b), which could explain the high stability of the HA-like fraction in maize straw biochar. According to Trompowsky et al. (2005), the higher aromatic C and lower O-alkyl C in the HA-like fraction from maize straw biochar might derive from the conversion of maize straw into refractory polycondensed aromatic structures during pyrolysis. On the other hand, Spaccini et al. (2000) observed that labile biomolecules (e.g., aliphatic compounds, polysaccharides and peptides) released during the decomposition of maize straw could be incorporated by chemical protection into native soil humic substances, which could help also explain why humic C concentrations increased in maize straw-amended soil. Although the same study has not been conducted in maize straw biochar-amended soil, we believed that labile biomolecules could also be partly incorporated into native soil humic substances during decomposition of maize straw biochar due to the similar chemical compositions of the two materials (Zhang et al., 2015a,b). In a recent study, Zhao et al. (2018b) speculated that easily decomposed aliphatic compounds in biochar from apple tree branches might participate in the humification process of soil.

The ratios of HAC/FAC and (HAC + FAC)/HUC could provide information on the relative rates of the formation and accumulation of different humic substance fractions (Li et al., 2016; Wang et al., 2016). The results from the present study (Fig. 2c and d) suggested that the application of maize straw and its biochar were more favorable for the formation of HA fractions compared to the CK treatment, whereas the same treatment did not significantly impact the relative rates of the formation of humic substance fractions compared with mineral fertilizers alone. Differing from our present results, Zhang et al. (2011) showed that the HAC/FAC ratio significantly increased after a seven-year of application of maize straw in combination with mineral fertilizers compared to mineral fertilizers alone in an Inceptisol. This discrepancy was not surprising because of the complexity of key factors, such as temperature, moisture and soil properties, that impacted the formation of humic substances in different ecosystems (Kononova, 1984; Stevenson, 1994; Tardy et al., 1997).

4.3. Effects of maize straw and its biochar application on soil aggregate stability

Among all of the treatments, treatments with maize straw and its biochar application generally contained higher proportion of macroaggregates and lower proportions of microaggregates and silt + clay fractions (Fig. 3a). This suggested that the application of maize straw and its biochar caused smaller microaggregates and silt + clay fractions to become cemented into larger macroaggregates. The aggregation process was more apparent for maize straw biochar than for maize straw, although the aggregate size distribution was not significantly different between the two treatments (Fig. 3a). The cementing agents involved in the formation of macroaggregates could be soil organic and humic C, aromatic groups, etc. (see below for additional details). On the other hand, the different sizes of macroaggregates contributed distinct for the increase of the proportion of macroaggregates. The ratio of large- to small-macroaggregates (Fig. 3c) revealed that large macroaggregates contributed more to the increase of the proportion of macroaggregates. In a three-year field microplot experiment, Ma et al. (2016) reported that the application of maize straw and its biochar, especially for maize straw biochar, significantly increased the proportion of large macroaggregates in a Mollisol, which was consistent with our results.

There were contrasting effects of biochar on soil aggregate stability under field conditions. Previous studies found that biochar amendment significantly increased (Liu et al., 2014; Ma et al., 2016; Are et al., 2017; Du et al., 2017; Li et al., 2017a,b,c; Zhang et al., 2017; Pituello et al., 2018), decreased (Liu et al., 2014), or had no effect (Liu et al., 2014; Jeffery et al., 2015; Zhang et al., 2015a,b; Dong et al., 2016; Zhang et al., 2017; Zhou et al., 2018) on soil aggregate stability in field experiments, depending on many factors such as biochar types, biochar dosages, and soil types, etc. In our present study, the variations in aggregate size distribution (Fig. 3a) led to a significant increase of MWD (Fig. 3d) after the application of maize straw biochar, suggesting that biochar had a positive effect on soil aggregate stability. In contrast, maize straw application did not significantly affect soil aggregate stability. There were several potential mechanisms that might bring about the increase of aggregate stability in biochar-amended soil. First, the enhancement of aggregate stability could be related to the high soil organic and humic C, which were the major organic cementing agents promoting the formation and stabilization of soil aggregates (Bronick and Lal, 2005; Spaccini et al., 2006). Second, the presence of abundant crop roots resulting from an increasing maize yield after biochar application (Fig. 1) could also contribute to the enhancement of soil aggregate stability by providing a variety of compounds (e.g.,

polysaccharide gums and polyvalent cations) that enmesh and realign soil fine particulates into more stable macroaggregates (Bronick and Lal, 2005; Hartley et al., 2016).

4.4. Effects of maize straw and its biochar application on organic and humic carbon concentrations in soil water-stable aggregates

As indicated in a previous study, biochar C could be incorporated into stable soil aggregates (Kelly et al., 2017). Thus, the application of maize straw biochar significantly increased organic C concentrations in all sizes of water-stable aggregates compared with the other treatments (Fig. 4a). Similar results have been observed by other researchers in biochar-amended soil (Du et al., 2017). In contrast, maize straw application generally exhibited little effect on soil aggregate-associated organic C concentrations (Fig. 4a). Our present findings supported the viewpoint of Wang et al. (2017) that biochar addition benefited soil C sequestration by improving soil aggregation and provided physical protection for C within soil aggregates. Thus, the contribution of biochar to soil C sequestration should not simply be attributed to the recalcitrant property of biochar itself (Wang et al., 2017). The changes in the ratio of macroaggregate- to microaggregate-associated organic C in biochar-amended soil (Fig. 4b) suggested that biochar C was preferentially incorporated into macroaggregates compared to microaggregate. The increase of C storage in macroaggregates favored the longer-term stabilization of C in stable microaggregates which form within macroaggregates (Six et al., 2002).

The C concentrations of humic substance fractions within different sizes of water-stable aggregates were also generally higher in treatments with than without maize straw and its biochar application, and with maize straw biochar than with maize straw application (Fig. 5a–c), suggesting that biochar C was also incorporated into humic substance fractions within stable soil aggregates. HU-like C accounted for approximately 96% of the total biochar C (Zhang et al., 2015a,b), which resulted in HUC concentration associated with different aggregate size fractions all being significantly higher in biochar-amended soil than in other treatments. Similar to organic C distribution (Fig. 4b), biochar C was also preferentially incorporated into the HU fraction within macroaggregates rather than that within microaggregates (Fig. 5f).

The change in the HAC/FAC ratio revealed that the application of maize straw biochar was more favorable to the formation of HA fractions with respect to FA in macroaggregates, particularly in large macroaggregates, whereas biochar application did not impact the relative rates of HA and FA fractions formation in microaggregates and silt + clay fractions (Fig. 5d). Thus, macroaggregates was the major contributor to the formation and accumulation of HA fractions in biochar-amended soil. Furthermore, the significant decline in the (HAC + FAC)/HUC ratio after the application of maize straw biochar (Fig. 5e) suggested that biochar was more favorable to the formation of insoluble humic substance fractions with respect to soluble humic substance fractions in different sizes of water-stable aggregates, which was consistent with the significantly higher HUC concentration associated with different aggregate size fractions in biochar-amended soil. Macroaggregates was also the major contributor to the formation and accumulation of HU fractions in biochar-amended soil (Fig. 5f).

4.5. Effects of maize straw and its biochar application on chemical compositions of soil small water-stable macroaggregates and the derived humic acid fractions

The five-year maize straw application generally did not impact the chemical composition of soil small water-stable macroaggregates and its HA fractions, whereas maize straw biochar application clearly altered their chemical composition (Table 1). The chemical composition of maize straw biochar itself could be responsible to that of native soil with maize straw biochar amendment. According to Zhang et al. (2015), both maize straw biochar and its HA-like fraction contained a large proportion of aromatic C and a small proportion of O-alkyl C with respect to its raw feedstock, which could explain the increased aromatic C and decreased O-alkyl C for small water-stable macroaggregates and its HA fractions after maize straw biochar application into the soil. As a consequence, maize straw biochar application resulted in the molecular structure of small water-stable macroaggregates and its HA fractions in native soil becoming more stable, more hydrophobic and less aliphatic as suggested by the increases in the ratios of alkyl C/O-alkyl C and hydrophobic C/hydrophilic C and the decrease in the ratio of aliphatic C/aromatic C (Table 1). As indicated in the above discussion, the enhancement of the hydrophobicity of soil organic matter contributed to the higher aggregate stability in biochar-amended soil.

5. Conclusions

This five-year field experiment demonstrates that, compared with unfertilized control and mineral fertilizers alone, the application of maize straw and its biochar increased the proportion of macroaggregates and exhibited a higher MWD of water-stable aggregates. The concentrations of organic and humic C in bulk soil and aggregate fractions were also generally higher after the application of maize straw and its biochar. The treatment effects were more apparent for maize straw biochar than for maize straw application. Moreover, the application of maize straw biochar clearly decreased the proportions of alkyl C and O-alkyl C but increased the proportion of aromatic C of small macroaggregates and its HA fractions, resulting in the alkyl C/O-alkyl C and hydrophobic C/hydrophilic C ratios being higher whereas the aliphatic C/aromatic C ratio being lower compared to other treatments. Our findings suggested that the application of maize straw biochar enhanced soil aggregate stability and aggregate-associated organic and humic C concentrations. Moreover, the molecular structure of small macroaggregates and its HA fractions became more stable, hydrophobic and aromatic. Therefore, maize straw biochar application represents an effective strategy to mitigate climate change and improve soil quality in Mollisols of Northeast China. Further research is needed to confirm the present conclusions by determining the long-term responses of soil organic and humic C to biochar amendment.

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References

Adani, F., Ricca, G., 2004. The contribution of alkali soluble (humic acid-like) and unhydrolyzed-alkali soluble (core-humic acid-like) fractions extracted from maize plant to the formation of soil humic acid. Chemosphere 56, 13–22.
Agegnehu, G., Srivastava, A.K., Bird, M.I., 2017. The role of biochar and biochar-compost in improving soil quality and crop performance: a review. Appl. Soil Ecol. 119, 156–170.
Anikwe, M.A.N., 2010. Carbon storage in soils of Southeastern Nigeria under different management practices. Carbon Balance Manag. 5, 5.
Are, K.S., Adelana, A.O., Fademi, I.O., Aina, O.A., 2017. Improving physical properties of degraded soil: potential of poultry manure and biochar. Agric. Nat. Resour. 51, 454–462.
Batjes, N.H., 1996. Total carbon and nitrogen in the soils of the world. Eur. J. Soil Sci. 47, 151–163.
Bi, L., Yao, S., Zhang, B., 2015. Impacts of long-term chemical and organic fertilization on soil puddlability in subtropical China. Soil Till. Res. 152, 94–103.
Bongiovanni, M.D., Lobartini, J.C., 2006. Particulate organic matter, carbohydrate, humic acid contents in soil macro- and microaggregates as affected by cultivation. Geoderma 136, 660–665.
Bronick, C.J., Lal, R., 2005. Soil structure and management: a review. Geoderma 124, 3–22.
Brunetti, G., Plaza, C., Clapp, C.E., Senesi, N., 2007. Compositional and functional

features of humic acids from organic amendments and amended soils in Minnesota, USA. Soil Biol. Biochem. 39, 1355–1365.
Cambardella, C.A., Elliott, E.T., 1993. Carbon and nitrogen distribution in aggregates from cultivated and native grassland soils. Soil Sci. Soc. Am. J. 57, 1071–1076.
Chen, Y., Zhang, X., He, H., Xie, H., Yan, Y., Zhu, P., Ren, J., Wang, L., 2010. Carbon and nitrogen pools in different aggregates of a Chinese Mollisol as influenced by long-term fertilization. J. Soil Sediment 6, 1018–1026.
Chen, S., Xu, C., Yan, J., Zhang, X., Zhang, X., Wang, D., 2016. The influence of the type of crop residue on soil organic carbon fractions: an 11-year field study of rice-based cropping systems in southeast China. Agric. Ecosyst. Environ. 223, 261–269.
Ding, Y., Liu, Y., Liu, S., Huang, X., Li, Z., Tan, X., Zeng, G., Zhou, L., 2017. Potential benefits of biochar in agricultural soils: a review. Pedosphere 27, 645–661.
Dong, X., Guan, T., Li, G., Lin, Q., Zhao, X., 2016. Long-term effects of biochar amount on the content and composition of organic matter in soil aggregates under field conditions. J. Soils Sediments 16, 1481–1497.
Du, Z., Zhao, J., Wang, Y., Zhang, Q., 2017. Biochar addition drives soil aggregation and carbon sequestration in aggregate fractions from an intensive agricultural system. J. Soils Sediments 17, 581–589.
Fischer, D., Glaser, B., 2012. Synergisms between compost and biochar for sustainable soil amelioration. In: Kumar, S., Bharti, A. (Eds.), Management of Organic Waste. Intech, Rijeka, pp. 167–198.
Ghimire, R., Lamichhane, S., Acharya, B.S., Bista, P., Sainju, U.M., 2017. Tillage, crop residue, and nutrient management effects on soil organic carbon in rice-based cropping systems: a review. J. Integ. Agric. 2017 (16), 1–15.
Graber, E.R., Tsechansky, L., Mayzlish-Gati, E., Shema, R., Koltai, H., 2015. A humic substances product extracted from biochar reduces *Arabidopsis* root hair density and length under P-sufficient and P-starvation conditions. Plant Soil 395, 21–30.
Gupta Choudhury, S., Srivastava, S., Singh, R., Chaudhari, S.K., Sharma, D.K., Singh, S.K., Sarkar, D., 2014. Tillage and residue management effects on soil aggregation, organic carbon dynamics and yield attribute in rice-wheat cropping system under reclaimed sodic soil. Soil Till. Res. 136, 76–83.
Han, L., Sun, K., Jin, J., Xing, B., 2016. Some concepts of soil organic carbon characteristics and mineral interaction from a review of literature. Soil Biol. Biochem. 94, 107–121.
Hartley, W., Riby, P., Waterson, J., 2016. Effects of three different biochars on aggregate stability, organic carbon mobility and micronutrient bioavailability. J. Environ. Manage. 181, 770–778.
Hayes, M.H.B., 2013. Relationships between biochar and soil humic substances. In: Xu, J., Wu, J., He, Y. (Eds.), Functions of Natural Organic Matter in Changing Environment. Springer, Dordrecht, pp. 959–963.
IPCC, 2014. Climate Change 2014: Synthesis Report. Contribution of Working Group I, II, and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. IPCC, Geneva, Switzerland.
Jeffery, S., Meinders, M.B.J., Stoof, C.R., Bezemer, T.M., van de Voorde, T.F.J., Mommer, L., van Groenigen, J.W., 2015. Biochar application does not improve the soil hydrological function of a sandy soil. Geoderma 251–252, 47–54.
Kelly, C.N., Benjamin, J., Calderón, F.C., Mikha, M.M., Rutherford, D.W., Rostad, C.E., 2017. Incorporation of biochar carbon into stable soil aggregates: the role of clay mineralogy and other soil characteristics. Pedosphere 27, 694–704.
Kononova, M.M., 1984. Organic matter and soil fertility. Sov. Soil Sci. 4, 71–86.
Lal, R., 1997. Residue management, conservation tillage and restoration for mitigating greenhouse effect by CO_2-enrichment. Soil Till. Res. 43, 81–107.
Lal, R., Lorenz, K., Hüttl, R., Schneider, B., von Braun, J., 2013. Ecosystem Services and Carbon Sequestration in the Biosphere. Springer, Dordrecht.
Li, Y., Zhang, J., Chang, S.X., Jiang, P., Zhou, G., Shen, Z., Wu, J., Lin, L., Wang, Z., Shen, M., 2014. Converting native shrub forests to Chinese chestnut plantations and subsequent intensive management affected soil C and N pools. For. Ecol. Manage. 312, 161–169.
Li, C., Gao, S., Zhang, J., Zhao, L., Wang, L., 2016. Moisture effect on soil humus characteristics in a laboratory incubation experiment. Soil Water Res. 11, 37–43.
Li, C., Cao, Z., Chang, J., Zhang, Y., Zhu, G., Zong, N., He, Y., Zhang, J., He, N., 2017a. Elevational gradient affect functional fractions of soil organic carbon and aggregates stability in a Tibetan alpine meadow. Catena 156, 139–148.
Li, Q., Jin, Z., Chen, X., Jing, Y., Huang, Q., Zhang, J., 2017b. Effects of biochar on aggregate characteristics of upland red soil in subtropical China. Environ. Earth Sci. 76, 372.
Li, W., Liu, M., Jiang, C., Wu, M., Chen, X., Ma, X., Li, Z., 2017c. Changes in soil aggregate-associated enzyme activities and nutrients under long-term chemical fertilizer applications in a phosphorus-limited paddy soil. Soil Use Manag. 33, 25–33.
Liu, Z., Chen, X., Yan, Jing, Li, Q., Zhang, J., Huang, Q., 2014. Effects of biochar amendment on rapeseed and sweet potato yields and water stable aggregate in upland red soil. Catena 123, 45–51.
Lou, Y., Xu, M., Wang, W., Sun, X., Zhao, K., 2011. Return rate of straw residue affects soil organic C sequestration by chemical fertilization. Soil Till. Res. 113, 70–73.
Lu, R., 2000. Analytical Methods for Soil Agricultural Chemistry. China Agricultural Science and Technology Press, Beijing.
Ma, N., Zhang, L., Zhang, Y., Yang, L., Yu, C., Yin, G., Doane, T.A., Wu, Z., Zhu, P., Ma, X., 2016. Biochar improves soil aggregate stability and water availability in a Mollisol after three years of field application. PLoS One 11, e0154091.
Mikha, M.M., Rice, C.W., 2004. Tillage and manure effects on soil and aggregate-associated carbon and nitrogen. Soil Sci. Soc. Am. J. 68, 809–816.
Mohammadi, A., Cowie, A., Mai, T.L.A., de la Rosa, R.A., Kristiansen, P., Brandão, M., Joseph, S., 2016. Biochar use for climate-change mitigation in rice cropping systems. J. Clean. Prod. 116, 61–70.
Peng, X., Yan, X., Zhou, H., Zhang, Y., Sun, H., 2015. Assessing the contributions of sesquioxides and soil organic matter to aggregation in an Ultisol under long-term fertilization. Soil Till. Res. 146, 89–98.
Pituello, C., Dal Ferro, N., Francioso, O., Simonetti, G., Berti, A., Piccoli, I., Pisi, A., Morari, F., 2018. Effects of biochar on the dynamics of aggregate stability in clay and sandy loam soils. Eur. J. Soil Sci. 69, 827–842.
Schmidt, M.W.I., Knicker, H., Hatcher, P.G., Kögel-Knabner, I., 1997. Improvement of ^{13}C and ^{15}N CPMAS NMR spectra of bulk soils, particle size fractions and organic material by treatment with 10% hydrofluoric acid. Eur. J. Soil Sci. 48, 319–328.
Senesi, N., Plaza, C., Brunetti, G., Polo, A., 2007. A comparative survey of recent results on humic-like fractions in organic amendments and effects on native soil humic substances. Soil Biol. Biochem. 39, 1244–1262.
Shi, Z., Jia, T., Wang, Y., Wang, J., Sun, R., Wang, F., Li, X., Bi, Y., 2017. Comprehensive utilization state of crop straw and estimation of carbon from burning in China. Chin. J. Agric. Resour. Reg. Plan. 38, 32–37 (in Chinese with English abstract).
Six, J., Conant, R.T., Paul, E.A., Paustian, K., 2002. Stabilization mechanisms of soil organic matter: implications for C-saturation of soils. Plant Soil 241, 155–176.
Spaccini, R., Piccolo, A., Haberhauer, G., Gerzabek, M.H., 2000. Transformation of organic matter from maize residues into labile and humic fractions of three European soils as revealed by ^{13}C distribution and CPMAS-NMR spectra. Eur. J. Soil Sci. 51, 583–594.
Spaccini, R., Mbagwu, J.S.C., Conte, P., Piccolo, A., 2006. Changes of humic substances characteristics from forested to cultivated soils in Ethiopia. Geoderma 132, 9–19.
Stevenson, F.J., 1994. Humus Chemistry: Genesis, Composition, Reactions, 2nd edn. Wiley, New York.
Stockmann, U., Adams, M.A., Crawford, J.W., Field, D.J., Henakaarchchi, N., Jenkins, M., Minasny, B., McBratney, A.B., Courcelles, Vd.Rd., Singh, K., Wheeler, I., Abbott, L., Angers, D.A., Baldock, J., Bird, M., Brookes, P.C., Chenu, C., Jastrow, J.D., Lal, R., Lehmann, J., O'Donnell, A.G., Parton, W.J., Whitehead, D., Zimmermann, M., 2013. The knowns, known unknowns and unknowns of sequestration of soil organic carbon. Agric. Ecosyst. Environ. 164, 80–99.
Tan, Z., Lin, C.S.K., Ji, X., Rainey, T.J., 2017. Returning biochar to fields: a review. Appl. Soil Ecol. 116, 1–11.
Tardy, Y., Schaul, R., Duplay, J., 1997. Thermodynamic stability fields of humus, microflora and plants. C. R. Acad. Sci. Paris 324, 969–976.
Trompowsky, P.M., de Melo Benites, V., Madari, B.E., Pimenta, A.S., Hockaday, W.C., Hatcher, P.G., 2005. Characterization of humic like substances obtained by chemical oxidation of eucalyptus charcoal. Org. Chem. 36, 1480–1489.
Wang, W., Chen, W., Wang, K., Xie, X., Yin, C., Chen, A., 2011. Effects of long-term fertilization on the distribution of carbon, nitrogen and phosphorus in water-stable aggregates in paddy soil. Agric. Sci. China 10, 1932–1940.
Wang, X., Peng, L., Zhang, X., Yin, G., Zhao, C., Piao, S., 2014. Divergence of climate impacts on maize yield in Northeast China. Agric. Ecosyst. Environ. 196, 51–58.
Wang, Y., Gao, S., Li, C., Zhang, J., Wang, L., 2016. Effects of temperature on soil organic carbon fractions contents, aggregate stability, and structural characteristics of humic substances in a Mollisol. J. Soil Sediment 16, 1849–1857.
Wang, D., Fonte, S.J., Parikh, S.J., Six, J., Scow, K.M., 2017. Biochar additions can enhance soil structure and the physical stabilization of C in aggregates. Geoderma 303, 110–117.
Xie, Z., Xu, Y., Liu, G., Liu, Q., Zhu, J., Tu, C., Amonette, J.E., Cadisch, G., Yong, J.W.H., Hu, S., 2013. Impact of biochar application on nitrogen nutrition of rice, greenhouse-gas emissions and soil organic carbon dynamics in two paddy soils of China. Plant Soil 370, 527–540.
Yang, X., Zhang, X., Fang, H., Zhu, P., Ren, J., Wang, L., 2003. Long-term effects of fertilization on soil organic carbon changes in continuous corn of northeast china: RothC model simulations. Environ. Manage. 32, 459–465.
Yang, X., Meng, J., Lan, Y., Chen, W., Yang, T., Yuan, J., Liu, S., Han, J., 2017. Effects of maize stover and its biochar on soil CO_2 emissions and labile organic carbon fractions in Northeast China. Agric. Ecosyst. Environ. 240, 24–31.
Zhang, J., Wang, L., Li, C., 2010. Humus characteristics after maize residues degradation in soil amended with different copper concentrations. Plant Soil Environ. 56, 120–124.
Zhang, J., Hu, F., Li, H., Gao, Q., Song, X., Ke, X., Wang, L., 2011. Effects of earthworm activity on humus composition and humic acid characteristics of soil in a maize residue amended rice-wheat rotation agroecosystem. Appl. Soil Ecol. 51, 1–8.
Zhang, J., Cai, H., Zhang, C., Ren, J., Wang, L., 2015a. First characterization of humic-like substances isolated from maize straw biochar. Fresen. Environ. Bull. 24, 1815–1821.
Zhang, Q., Du, Z., Lou, Y., He, X., 2015b. A one-year short-term biochar application improved carbon accumulation in large macroaggregate fractions. Catena 127, 26–31.
Zhang, S., Wang, R., Yang, X., Sun, B., Li, Q., 2016. Soil aggregation and aggregating agents as affected by long term contrasting management of an Anthrosol. Sci. Rep. 6, 39107.
Zhang, M., Cheng, G., Feng, H., Sun, B., Zhao, Y., Chen, H., Chen, J., Dyck, M., Wang, X., Zhang, J., Zhang, A., 2017. Effects of straw and biochar amendments on aggregate stability, soil organic carbon, and enzyme activities in the Loess Plateau, China. Environ. Sci. Pollut. Res. 24, 10108–10120.
Zhao, H., Shar, A.G., Li, S., Chen, Y., Shi, J., Zhang, X., Tian, X., 2018a. Effect of straw return mode on soil aggregation and aggregate carbon content in an annual maize-wheat double cropping system. Soil Till. Res. 175, 178–186.
Zhao, S., Ta, N., Li, Z., Yang, Y., Zhang, X., Liu, D., Zhang, A., Wang, X., 2018b. Varying pyrolysis temperature impacts application effects of biochar on soil labile organic carbon and humic fractions. Appl. Soil Ecol. 123, 484–493.
Zhou, H., Fang, H., Zhang, Q., Wang, Q., Chen, C., Mooney, S.J., Peng, X., Du, Z., 2018. Biochar enhances soil hydraulic function but not soil aggregation in a sandy loam. Eur. J. Soil Sci. https://doi.org/10.1111/ejss.12732.
Zhu, L., Xiao, Q., Shen, Y., Li, S., 2017. Microbial functional diversity responses to 2 years since biochar application in silt-loam soils on the Loess Plateau. Ecotox. Environ. Saf. 144, 578–584.